OPERATIONAL COMPONENTS AND DESIGN OF CONSTRUCTED WETLANDS USED FOR TREATMENT OF SWINE WASTEWATER

P.G. Hunt, M. E. Poach, A.A. Szogi, G. B. Reddy, K. C. Stone, F. J. Humenik, and M. B. Vanotti[1]**.**

ABSTRACT

Constructed wetlands are a natural and passive treatment method for swine wastewaters. We have investigated swine lagoon wastewater treatment in both continuous marsh and marsh-pond-marsh (MPM) type constructed wetlands for their N and P treatment efficiency, ammonia volatilization, denitrification, and treatment system design. Neither type of wetland system was effective in the removing large quantities of P. Continuous marsh systems were able to remove more N than the MPM systems, particularly if planted to rushes/bulrushes (*Juncus effusus, Scirpus validus, Scirpus americanus, Scirpus cyperinus*). Plant and soil accumulations of N and P were important at very low loading rates; but as the loading rates exceeded 5 kg ha^{-1}day^{-1}, They became a small part of the removal process. Although, ammonia volatilization was present; it was generally <10% of the applied N in the marsh sections, and it was highly correlated to nitrogen concentration. However, the pond sections of the MPM systems had high levels of ammonia volatilization when loading rates exceeded 15 kg N ha^{-1} day^{-1}. Water depth had a large impact on denitrification, as did the plant cover. Treatment efficiency was reasonably predicted by current modeling techniques used for municipal wastewater treatment in constructed wetlands.

KEYWORDS. Denitrification, Water depth, Ammonia volatilization, Plant nutrient uptake, Soil accumulation, Design parameters.

INTRODUCTION

Throughout the world, it is common for swine production enterprises to initially treat wastewater in anaerobic lagoons and subsequently apply the treated wastewater to land. This method is satisfactory when large tracts of cropland are available, sensitive ecosystems are absent, and neighbors are tolerant (Stone et al., 1995). However, these conditions often do not exist, and superior treatment alternatives are needed. One of these alternatives is constructed wetlands.

Wetlands have been used successfully for advanced treatment of municipal and residential wastewaters in the USA and around the world for over three decades (Hammer, 1989; Kadlec and Knight, 1996). They are considered to be both natural and passive wastewater treatments. Their function and reliability for animal wastewater treatment have been documented, but much remains to be learned about this technology (Hunt et al., 1999; Knight et al., 2000; Hunt and Poach, 2001; Hunt et al., 2002a).

Generally, the focus of animal wastewater treatment in constructed wetlands is to remove nutrients and; thereby, decrease the land necessary to receive, transform, and assimilate the

[1] Authors are **P.G. Hunt, M.E. Poach, A.A. Szogi, K.C. Stone,** and **M.B. Vanotti**, USDA-ARS, Coastal Plains Soil, Water, and Plant Research Center, 2611 W. Lucas St., Florence, SC 29501, **G.B. Reddy**, Department of Natural Resources, NC A&T State Univ., Greensboro, NC 27411; and **F.J. Humenik**, CALS Animal Waste Management Program, NC State Univ., Raleigh, NC 27695.

remaining nutrients in the wastewater (Hunt et al., 1995; Knight et al., 2000). Land application is necessary because direct discharge of animal wastewater is not permitted even after treatment. The objectives of this paper are to present findings and ideas on constructed wetland treatment of swine wastewater, specifically 1) mass removal of N and P, 2) soil and plant accumulation, 3) ammonia volatilization, 4) denitrification, and 5) design considerations.

Experimental Sites

Investigations discussed in this paper were conducted from 1992 to present at experimental wetland sites in North Carolina. The continuous marsh site was in the eastern coastal plain in Duplin County and the marsh-pond-marsh site was in Greensboro at North Carolina A&T State University. The site description and operational procedures were reported in Reddy et al., 2000; Hunt et al., 2002a; and Poach et al. 2002.

DISCUSSION

Mass Removal of N and P

When we analyzed the continuous marsh wetlands on a monthly basis, we found that substantial removal of N was accomplished over a considerable range of mean loading rates (3 to 40 kg N ha^{-1} day^{-1}) by both the bulrush and bur-reed/cattails (*Sparganium americanum, Typha latifolia, Typha angustifolia*) wetlands (Hunt et al., 2002a). Regression equations of monthly mean N load versus N removal for the bulrush and cattail wetlands were: N removal = 0.73 N load + 1.39, R^2 = 0.94 and N removal = 0.85 N load + 0.18, R^2 = 0.93; respectively. The wetlands were less effective in N removal when loaded at > 25 kg N ha^{-1} day^{-1}. However, removals of applied N were always > 50% and most were > 75%. The marsh-pond-marsh wetlands were less effective. On annual nitrogen loading rates from 5 to 37 kg N ha^{-1} day^{-1}, the wetlands removed > 45% of the nitrogen from the wastewater (Figure 1).

The actual N loading rates and treatment efficiencies for these wetlands are both very high relative to traditional land application treatment and consistent with other wetland literature. For instance, in Alabama, McCaskey et al. (1994) found 99 to 82% removal of total N from swine lagoon wastewater treated with constructed wetlands that were loaded at 2.5 to 12.5 kg N ha^{-1} day^{-1}. Our results also correspond with those reported by Cathcart et al. (1994) for a marsh-pond-marsh (MPM) constructed wetland system in Mississippi; they obtained mass ammonia-N reductions of 71% when their system was loaded with 14 kg N ha^{-1} day^{-1}.

After wetlands have dramatically reduced the quantity of N in the wastewater, much less cropland will be required to accept the N load. Moreover, the timing of the applications can be managed more easily to accommodate both weather patterns and crop needs. Each hectare of wetland could remove >3.5 Mg N each year with three conditions: 1) loading rate of 20 kg N ha^{-1} day^{-1}, 2) 70% N removal, and 3) 250 days of wetland operation. When sufficient land is not available for assimilation of wastewater N at agronomic rates (< 0.5 Mg^{-1} ha^{-1} yr^{-1}) or expansion of the operation is desired, constructed wetlands can offer a feasible alternative for managing the N load from swine facilities. Additionally, wetland systems are operationally passive, and they cycle N via natural processes.

Neither the bulrush nor cattail wetlands were consistently effective in the mass removal of P, and both systems were generally < 50% effective when the loading rates exceeded 4 kg P ha^{-1} day^{-1} (Hunt et al., 2002a). There was modest correlation of P load and removal [(cattail wetland, P removal = 0.50 P load - 0.15; R^2 = 0.48) (bulrush wetland, P removal = 0.31 P load + 0.33; R^2 = 0.35)]. The low P removals are consistent with the expectation based on both the reduced Eh conditions of the wetland soil and other reports of P treatment efficiency (Hunt and Poach, 2001; Knight et al., 2000; and Szögi et al., 2001).

Plant Growth and Nutrient Accumulations

During one five-year period of these investigations, there was a substantial range of annual plant dry matter accumulation, 11 to 28 Mg ha^{-1} yr^{-1}; bulrushes and cattails alternate in which had the highest dry matter. Causes of this large range included yearly variable community composition and insect-disease pressure. However, the five-year means were very similar for the bulrush and cattail wetlands, 17.5 and 17.2 Mg ha^{-1}, respectively (table 3). Moreover, these means are consistent with other wetland systems (DeBusk and Ryther, 1987).

Plant N annual accumulations ranged from 114 to 595 kg ha^{-1} yr^{-1}. However, the mean annual accumulations were not significantly different for the bulrush and cattail wetlands, 354 and 317 kg ha^{-1} yr^{-1}, respectively. Plant P annual accumulations ranged from 20 to 206 kg ha^{-1} yr^{-1}. As with N, the mean annual accumulations of P were not significantly different for the bulrush and cattail wetlands, 76 and 83 kg ha^{-1} yr^{-1}, respectively. At the low loading rates, the N and P annually accumulated by the plants were a significant component of the wetland's annual nutrient budget (~ 30 and 38%, respectively). However, once the application rate exceeded 10 kg N ha^{-1} day^{-1}, plant accumulation of N and P was a minor component (< 3%).

Nonetheless, accumulation of plant dry matter and uptake of nutrients were very important. The plant dry matter provided for nutrient storage via internal cycling in the wetland. Since plant dry matter was not harvested, it accumulated on the wetland bottom after the plants had aged and their aerial parts succumbed to frost. This accumulation allowed a significant litter layer to establish and function as both a source of carbon and an extensive reaction surface for microorganisms. In particular, the carbon exuded from the roots along with the carbon in the dead plant litter provided the energy necessary to drive the denitrification process. This may be particularly critical if high rates of nitrified wastewater were to be added. Hunt et al. (1999) reported the advantage of plant litter for denitrification when ~ 50 kg N ha^{-1} day^{-1} were added to wetland microcosms.

Soil Accumulations of N and P

Physical and chemical processes of the wetland promoted N and P accumulations in both the litter layer and mineral soil. Thus, we anticipated soil accumulations of N and P (Szögi and Hunt, 2001; Szögi et al., 2000). Mean N accumulation for the wetland systems had reached 1027 kg ha^{-1} during the final year (table 2). Yet, this accumulation was relatively small (< 10%) compared to the > 18 Mg of N applied during a five-year period. Mean N accumulations were not significantly different between the bulrush and cattail wetland any year.

The P did not accumulate in the wetlands over the study period (table 5). In fact, it actually decreased in the cattail wetland. Correlations of P accumulation and distance from the wetland inlet were low ($R^2 < 0.17$). These findings are consistent with the finite soil P adsorption and the generally reduced Eh conditions of the wetland soils, especially in the cattail wetlands. In contrast to N, P was present in the effluent in significant quantities. This was an expected result because these wetlands were loaded with large amounts of P and the reductive environment could promote P solubility. When P loads are high, some form of treatment augmentation will be necessary to obtain low levels of P in the effluent (Lee et al., 1976; Davies et al., 1993; Vanotti et al. 2003; Poach et al. 2003).

Ammonia Volatilization

Ammonia (NH_3) volatilization has been implicated as a significant N removal mechanism operating in treatment wetlands when wastewater ammonia is greater than 20 mg L^{-1} (Payne and Knight, 1997). Animal wastewater generally has ammonia concentrations > 20 mg L^{-1}, high oxygen demand, and very little nitrogen in the nitrate form (Kadlec and Knight, 1996; Knight et al., 2000). To determine the contribution of volatilization to nitrogen removal, a special open-ended enclosure with forced airflow was used to measure NH_3 volatilization from 1- x 4-m plots in both continuous marsh and MPM wetlands. Poach et al. (2002) gave a detailed description of the method used to measure NH_3 volatilization.

Ammonia volatilized from continuous marsh wetlands when they received liquid swine manure, but the volatilization did not account for a major portion of N removed by the wetlands (Table 3; Poach et al., 2002; Poach et al. 2003). Ammonia volatilization accounted for < 15% of the nitrogen load to the wetlands. Preliminary research on MPM wetlands indicated that NH_3 volatilization becomes a concern at N loads > 15 kg N ha^{-1} day^{-1} (Hunt et al., 2002b). The pond sections contributed the bulk of the overall ammonia volatilized from each system at these N loads. Ammonia-N volatilization rates from all the marsh sections were similar to NH_3-N volatilization rates exhibited by continuous marsh wetlands. A combination of biological and physical factors likely contributed to the high NH_3-N volatilization rates exhibited by the ponds. Pond sections which received N loads > 15 kg ha^{-1} day^{-1} supported an algal community, while pond sections which received N loads < 12 kg ha^{-1} day^{-1}) supported duckweed. Algal photosynthesis can lead to a rise in pH during the day, which can increase NH_3 volatilization. Ponds also had high wind exposed surface area, which can also increase NH_3 volatilization. These results indicate that at high N loads continuous marshes may be better suited for treating swine wastewater than MPM systems. Also, while NH_3 volatilization was a minor nitrogen removal mechanism in the marshes, its contribution to nitrogen loss should not be ignored.

Further research on the continuous marsh wetlands indicated that NH_3 volatilization was reduced when nitrification was used to lower the ammoniacal nitrogen in the liquid manure before wetland application (Poach et al., 2003). Pre-wetland nitrification also enhanced the removal of N by the process of denitrification because denitrification, which is the preferred N removal mechanism, is limited by the ability of the constructed wetland to convert ammonia to nitrate/nitrite (Hunt et al., 2003).

Denitrification

Denitrification was determined to be the predominant mechanism for N removal. This conclusion was reached by mass balance difference and the measure of denitrification potential via the acetylene blockage method (Hunt et al., 2002a). In the investigation of denitrification in these wetlands, they found that the bulrush wetlands had higher denitrification enzyme activity (DEA) means (P #0.001) than did cattail wetlands; 0.516 and 0.210 µg N g^{-1} soil hr^{-1}, respectively. When converted to an area basis, the mean value for the disturbed samples of the bulrush wetlands was equivalent to 9.55 kg N ha^{-1} day^{-1}. DEA rates increased over time as the rate of applied N increased and the wetlands matured. DEA in the control treatment was well correlated to the cumulative total N applied to both the bulrush and cattails, $r^2 = 0.73$ and 0.62, respectively. Nitrate was essentially not present in the wetland. Accordingly, nitrate was generally the limiting factor in the DEA measures, especially in the bulrush wetlands. On the other hand, carbon provided by the wetland plants and the wastewater was generally sufficient unless high additions of nitrate were made. Water depth was a very significant factor in the control of DEA in the bulrush wetlands. In bulrush wetlands, the slope and r^2 values of the control treatment were -0.013 µg N g^{-1} h^{-1} mm^{-1} depth and $r^2 = 0.89$, respectively. Furthermore, the effect of depth on DEA in the bulrush wetlands was extraordinarily consistent among all treatments (Fig 2 and 3). The slopes varied by only 0.001 µg N g^{-1} h^{-1} mm^{-1} depth, and the r^2 values ranged from 0.75 to 0.99. Decreased DEA with depth was likely caused by decreased O_2 and Eh of the effluent as well as the increased diffusion path associated with the greater water depth. Cattail wetlands were very different than bulrush wetlands in relation to water depth and denitrification. In the control treatment, there was very little effect of water depth on denitrification. The slope was -0.002 µg N g^{-1} h^{-1} mm^{-1}, and the r^2 was 0.82. The lack of response to water depth was likely because the cattails were not able to establish oxidative conditions sufficient for nitrification even with the relatively more oxidized condition associated with the shallower depth. This conclusion is supported by the fact that addition of nitrate and C to cattail wetlands produced DEA responses to depth much more similar to bulrush wetlands.

Denitrification in wetlands is generally associated with microbes in the soil and/or detritus layer. However, denitrification in the floating sludge layer was much (>20-fold) higher on a unit weight basis than in the soil and/or detritus layers. The rates of denitrification in the floating sludge were similar to those we obtained with polyvinyl alcohol-immobilized-denitrifying sludge pellets (Hunt et al., 2002b). These data indicate the very high potential for wetland removal of nitrate-N. However, at some level, available carbon for microbial respiration would be the limiting factor, and carbon would need to be added to the wetland (Hunt et al., 2002).

Wetland Design Parameters

Technical requirements for wetland design have been based mainly on municipal systems and limited data on animal waste systems. Design guidelines for sizing wetlands are typically based on a first-order kinetics area-based uptake model (Reed et al., 1995 and Kadlec and Knight, 1996). Stone et al. (2002) used the Kadlec and Knight (1996) model to calculate rate constants for nitrogen treatment in a continuous surface flow constructed wetland in Duplin County, NC. This model incorporates the hydraulic loading rate, concentrations into and out of the wetlands, and a temperature-based rate constant. Many of the literature-cited rate constants were calculated from summary data from various wetlands. Stone et al. (2002) looked at detailed treatment and performance data covering several years and loading rates. Nonetheless, calculated rate constants were generally similar to or slightly lower than those reported in the limited literature. Using calculated rate constants would result in a slightly more conservative constructed wetland design. A newly constructed wetland with mean loading rates and concentrations similar to the wetland system discussed in this paper would be slightly larger (~5%) based on ammonia-N compared to those based on the currently available guidelines. In swine lagoon wastewater systems where total N is denominated by ammonia-N, this would be a very important consideration for wetland design.

CONCLUSION

1. In continuous marsh wetlands with sloped bottoms either cattail or bulrush wetlands removed > 60% of N at loading rates of < 25 kg N $ha^{-1}day^{-1}$, but marsh-pond-marsh wetlands with flat bottoms and cattails were less effective with > 45% N removal.
2. Neither system was effective in mass removal of P. Thus, constructed wetlands will have to be augmented to remove the high P content of swine wastewater.
3. Plant and soil accumulation were important at very low loading rates, but became minor factors for N or P removal at rates > 10 kg N ha^{-1} day^{-1}.
4. Although ammonia volatilization was present in the marsh section of the wetlands, it was not a predominant N loss factor. However, in the pond section of the MPM wetlands, ammonia volatilization was high.
5. Denitrification appears to be the predominant removal process. Bulrush wetlands had higher DEA means than did cattail wetlands, 0.516 and 0.210 μg N g^{-1} soil hr^{-1}, respectively. When converted to an area basis, the mean value for disturbed samples of the bulrush wetlands was equivalent to 9.55 kg N ha^{-1} day^{-1}.
6. Nitrate was generally the limiting factor for denitrification, especially in the bulrush wetlands. Conversely, carbon provided by wetland plants and wastewater was generally sufficient unless high additions of nitrate were made.
7. Water depth was a very significant factor in the control of DEA in the bulrush wetlands. Furthermore, the effect of depth on DEA in the bulrush wetlands was extraordinarily consistent among all treatments. The slopes varied by only 0.001 μg N g^{-1} h^{-1} mm^{-1} depth, and the r^2 values ranged from 0.75 to 0.99.
8. Cattail wetlands were very different than bulrush wetlands in relation to water depth and denitrification. However, with the addition of nitrate and C cattail wetlands produced DEA responses to depth much more similar to bulrush wetlands.
9. Calculated rate constants from these studies were generally similar to or slightly lower than those reported in the limited literature requiring about 5% larger design.

10. Finding of these studies support the philosophy that constructed wetlands are likely to work best when used as part of a total waste management system.

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Table 1. Plant dry matter, nitrogen, and phosphorus accumulation in constructed wetlands (modified from Hunt et al., 2002)

†	Bulrush.	Cattail	$LSD_{0.10}$	
Dry matter	17.5	17.2	NS	Mg ha^{-1} yr^{-1}
Nitrogen	354	317	NS	Kg ha^{-1} yr^{-1}
Phosphorus	76	83	NS	Kg ha^{-1} yr^{-1}

† Mean of five years

Table 2. Applied and soil accumulated nitrogen and phosphorus in constructed wetlands (modified from Hunt et al., 2002)

Nutrient	Bulrush		Cattail		Wetland differences
	Applied	Accumulated†	Applied	Accumulated	$LSD_{0.10}$
	kg ha^{-1}				
Nitrogen	19144	1057	18035	997	NS
Phosphorus	4043	250	3719	53	*

* Significant by Least Significant Difference at the 0.10 level.

† The accumulated values are the increase or decrease from the initial soil N and P content values of 453 and 551 kg ha^{-1}, respectively.

Table 3. Ammonia volatiliztion from constructed wetlands (modified from Poach et al., 2002)

Month	N Load	NH_3-N Volatilization	
	(kg ha^{-1} day^{-1})	(kg ha^{-1} day^{-1})	(% of load)
May	54	3.8	7
July	18	2.7	15

Table 4. Rate constant (K20), dimensionless temperature coefficient (θ), and background concentrations (C*) calculated simultaneously using the Excel solver routine (modified from Stone et al., 2002)

	K20, θ, C* Calculated			C* Assumed†			C* = 0	
	K20 (m yr^{-1})	θ	C* (mg L^{-1})	K20 (m yr^{-1})	θ	C* (mg L^{-1})	K20 (m yr^{-1})	θ
TN Bulrush/rush	8.85	1.02	10.99	8.71	1.02	10	7.45	1.03
TN Cattail/burreed	8.66	0.98	5.81	9.35	0.98	10	7.86	0.99
NH_4-N Bulrush/rush	8.98	1.03	7.73	8.60	1.03	3	7.82	1.03
NH_4-N Cattail/burreed	9.39	0.98	4.32	9.20	0.98	3	8.62	0.99

† C* assumed from Knight et al. (2000).

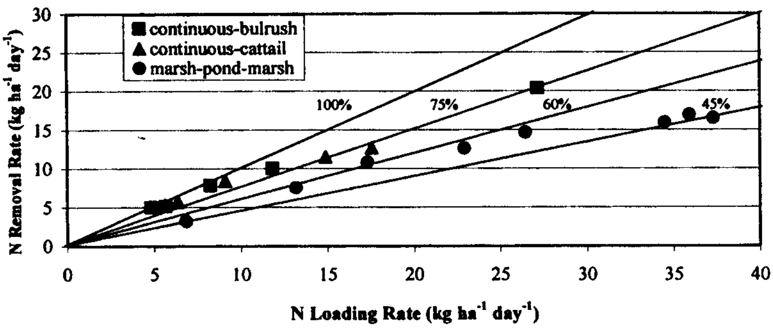

Figure 1. Comparison of different constructed wetland systems for annual removal of N vs. N loadings.

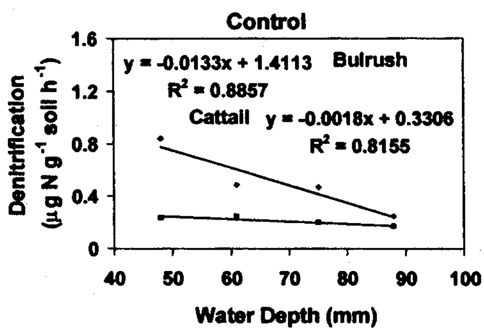

Figure 2. Impact of swine wastewater depth in constructed wetlands on denitrification (Hunt et al., 2003).

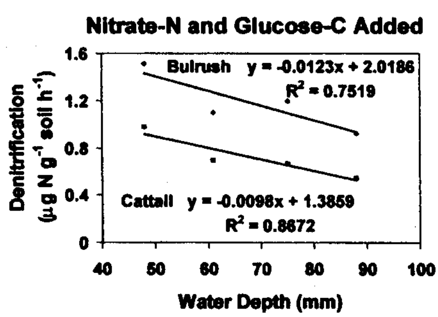

Figure 3. Impact of swine wastewater depth in constructed wetlands on denitrification as affected by addition of nitrate-N and glucose-C (Hunt et al., 2003).

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